

# Pre-Design Investigation and Baseline Sampling Data: EPA Evaluation Report

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## ACRONYMS

1,2,3,4,7,8-HxCDF	1,2,3,4,7,8-hexachlorodibenzofuran
1,2,3,7,8-PeCDD	1,2,3,7,8-pentachlorodibenzo- <i>p</i> -dioxin
2,3,4,7,8-PeCDF	2,3,4,7,8-pentachlorodibenzofuran
2,3,7,8-TCDD	2,3,7,8-tetrachlorodibenzo- <i>p</i> -dioxin
2,3,7,8-TCDF	2,3,7,8-tetrachlorodibenzofuran
3-D	three-dimensional
µg/kg	micrograms per kilogram
AECOM	AECOM Technical Services
ASAO	administrative settlement agreement and order on consent
ATSDR	Agency for Toxic Substances and Disease Registry
BEHP	bis(2-ethylhexyl)phthalate
BSAF	biota to sediment accumulation factor
cm	centimeters
COC	contaminant of concern
cPAH	carcinogenic polycyclic aromatic hydrocarbon
CSM	conceptual site model
CUL	cleanup level
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethene
DDT	dichlorodiphenyltrichloroethane
DDx	dichlorodiphenyltrichloroethane and its derivatives
dioxins/furans	polychlorinated dibenzo- <i>p</i> -dioxins and furans
D/U Reach	Downtown Reach and Upriver Reach
EPA	U.S. Environmental Protection Agency
ESD	explanation of significant differences
FS	feasibility study
FSP	field sampling plan
FWM	food web model
Geosyntec	Geosyntec Consultants, Inc.
GSI	GSI Water Solutions, Inc.
HHRA	human health risk assessment
Integral	Integral Consulting, Inc.
MC	Multnomah Channel
mg/L	milligrams per liter
MNR	monitored natural recovery
ODFW	Oregon Department of Fish and Wildlife
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PDI	pre-design investigation
PDI/BL	pre-design investigation and baseline sampling
Pre-RD Group	Pre-Remedial Design Group
PTW	principal threat waste
QA	quality assurance

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RAL	remedial action level
RI	remedial investigation
RM	river mile
ROD	record of decision
SDU	sediment decision unit
SIL	Swan Island Lagoon
site	Portland Harbor Superfund Site
SMA	sediment management area
SRS	stratified random sampling
SWAC	surface area weighted average concentration
TADT	technology application decision tree
TEQ	toxic equivalent
TSS	total suspended solids
UCL	upper confidence limit
USGS	U.S. Geological Survey
yr <sup>-1</sup>	per year

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## Purpose

From March 2018 to May 2019, the Portland Harbor Pre-Remedial Design Group (Pre-RD Group) performed the pre-design investigation and baseline sampling (hereinafter referred to as PDI/BL) under a U.S. Environmental Protection Agency (EPA) administrative settlement agreement and order on consent (ASAOC) and with EPA field oversight. The Pre-RD Group summarized their findings with the PDI/BL data in their *PDI Evaluation Report* (AECOM Technical Services [AECOM] and Geosyntec Consultants, Inc. [Geosyntec] 2019a). The EPA received the PDI/BL data from the Pre-RD Group between August 2018 (bathymetry survey data file) and July 2019 (12-month acoustic fish tracking study data download). As it received the PDI/BL data, the EPA performed data quality assurance (QA) reviews to assess the overall quality of the data. The EPA found that the PDI/BL data are complete, of suitable quality, and generally acceptable for inclusion into the Portland Harbor Superfund Site (site) database, pending minor corrections described in *EPA Review Comments on PDI Evaluation Report and Acoustic Fish Tracking Study 12-Month Addendum* (EPA 2019).

The EPA additionally performed spatial and statistical analyses to understand the PDI/BL data and how they agree with or indicate a change may be needed to the conceptual site model (CSM) summarized in the *Record of Decision* ([ROD]; EPA 2017a). These analyses were developed from October 2018 through September 2019 and informed the development of comments in *EPA Review Comments on PDI Evaluation Report and Acoustic Fish Tracking Study 12-Month Addendum* (EPA 2019). The EPA's analyses were initially summarized in a series of briefing memoranda and presentations developed from February to September 2019, which have been edited for consistency in formatting and are reproduced herein.

The EPA refers to the Pre-RD Group sampling effort and data interchangeably as PDI/BL and pre-design investigation (PDI) throughout this report.

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## 1. Surface Sediment Statistics

The June 2017 ASAOC states, “The Pre-RD Group proposes to conduct a comprehensive 2017/2018 synoptic sampling program of surface sediment, select sediment cores, fish tissue, surface water, background porewater, and bathymetry/fish tracking studies.”

Two types of surface sediment samples were collected in the site:

- Unbiased baseline stratified random sampling (SRS) within a grid system (for long-term monitoring) (n = 424 samples) (**Figure 1-1**)
- Targeted samples located in sediment management areas (SMAs) to support further refinement of the SMA footprints (n = 231 samples) (**Figure 1-2**)

Core (subsurface) samples were also collected in the SMAs (n = 90). Surface sediment samples were collected from the Downtown Reach (n = 29) and the background Upriver Reach (n = 30). This area is collectively referred to as the D/U Reach.

### 1.1. Are there significant differences?

Three aspects of this question will be reviewed.

- 1) Location (Have areas of contamination changed?)
- 2) Concentrations (Have contamination levels changed?)
- 3) Equivalence (How do the current site concentrations compare to the background Upriver Reach?)

#### 1.1.1. Location (Have areas of contamination changed?)

Table 21 of the ROD lists the 11 contaminants of concern (COCs) that drive the areas of active sediment remediation in the site. These COCs are broken down into the six focused COCs, which have remedial action levels (RALs), and the five additional COCs that are considered principal threat waste (PTW) and have PTW threshold concentrations. **Figure 1-3** shows the 2018 surface sediment samples that exceed RALs and/or PTW thresholds compared to the ROD’s SMA footprint. **Figures 1-4a** through **1-4e** show the same results compared to the ROD and *Proposed Explanation of Significant Differences* (ESD) SMA footprints (EPA 2018). **Figures 1-5** through **1-14** show the distribution of the ROD Table 21 COCs in surface sediment. Chlorobenzene was not included in the 2018 sampling.

Generally, most of the ROD surface sediment-based SMAs still have RAL exceedances. There have not been major shifts in the surface sediment hot spots, and the contaminated areas are generally still contaminated. In some areas, RAL exceedances are no longer present on the surface, whereas in others, RAL exceedances are present in areas not seen in the ROD. During full remedial design, higher density sampling will be needed to fully and appropriately delineate the SMAs in both the surface and subsurface sediments.

### 1.1.2. Concentrations (Have contamination levels changed?)

To address this question, it is necessary to focus on individual areas and COCs. Portland Harbor is a large site, where areas of contaminated sediment are interspersed among broad, relatively clean areas. As a result, sitewide averages can mask localized conditions and site-specific changes. Sediment decision units (SDUs) were used in the ROD to depict conditions in the most contaminated areas. The SDUs are approximately 1 mile in river length and are centered on the most contaminated areas.

An important caveat is that the analyses below depict evaluations of data from within the SRS grid cells (cell boundaries are shown in **Figures 1-1** and **1-2**). The SRS grid cell sizes range from 0.3 to 25 acres, and these comparisons are premised on the assumption that both the PDI/BL and remedial investigation (RI) and feasibility study (FS) samples are appropriate and comparable representations of that cell. That may not always be the case; however, the assumption is necessary if the RI/FS data are to be compared to the PDI/BL data. See Section 11 for further discussion on the differences between the PDI/BL and RI/FS sampling designs.

Surface sediment temporal change in the SDUs between the PDI/BL and RI/FS data was assessed using the Wilcoxon signed-rank test. This analysis was performed for the ROD Table 21 COCs. RI/FS samples in an SRS grid cell within the SDU were debiased by taking the median result and comparing to the median result of PDI/BL samples (SRS and SMA, where available). **Figure 1-15** summarizes the analysis and condenses the range of COCs into a single comparison in which gray SDUs have no statistical differences for the majority of COCs, red SDUs show significant increases, and green SDUs show significant decreases. Eight of the 13 SDUs showed overall nonsignificant change between the PDI/BL and RI/FS data, whereas 4 SDUs showed a significant increase, and 1 showed a significant decrease. These results suggest that the more contaminated areas within the site have experienced a negligible decrease in surface sediment COC concentrations. Increases are predominantly driven by polychlorinated dibenzo-*p*-dioxins and furans (dioxins/furans), which were evaluated in target areas during the RI/FS and sitewide during the PDI/BL.

**Figures 1-16a** through **1-16j** present the SDU and ROD Table 21 COC data in greater detail. In the box and whisker plots, the PDI/BL data are compared to the debiased RI/FS data for each SDU. The data are presented as a ratio (new/old) so that the degree of change can be readily discerned. Ratios less than 1.0 indicate declines in concentration, and boxes that overlap the red dashed line at 1.0 are indicative of little change. The box and whisker plots suggest that surface sediment concentrations for total polychlorinated biphenyls (PCBs) have decreased in most SDUs while total polycyclic aromatic hydrocarbons (PAHs), carcinogenic PAHs (cPAHs), dichlorodiphenyltrichloroethane and its derivatives (DDx), and naphthalene show nonsignificant change. Similar to the results of the Wilcoxon signed-rank test, the dioxin/furan congeners with ROD Table 17 cleanup levels (CULs), 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD); 1,2,3,7,8-pentachlorodibenzo-*p*-dioxin (1,2,3,7,8-PeCDD); 2,3,4,7,8-pentachlorodibenzofuran (2,3,4,7,8-PeCDF); 2,3,7,8-tetrachlorodibenzofuran (2,3,7,8-TCDF); and 1,2,3,4,7,8-hexachlorodibenzofuran (1,2,3,4,7,8-HxCDF) predominantly show increases in concentration from the RI/FS to the PDI/BL data.

**Figures 1-17a** through **1-17j** compare the unbiased PDI/BL data (i.e., SRS only) to the RI/FS data for each of the 428 SRS grid cells in the 2018 sampling grid (not all cells had RI/FS data for comparison). Each figure in the series presents the regression analysis for a different ROD Table 21

COC. Gray squares represent the median (50th percentile) of the RI/FS samples. Data points in the upper right quadrant of each figure are the cells above RALs for both the RI/FS and PDI/BL data. The points above the 1 to 1 line are SRS grid cells where the surface sediment COC concentration has increased from the RI/FS to the PDI/BL data. Points below the line are cells where the concentration decreased. Percentages of the cells that increased or decreased are presented in the figures (sitewide is black text; RAL exceedance grid cells are red text). **Figures 1-17a through 1-17j** show that sitewide concentrations of total PCBs, total PAHs, total DDX, and naphthalene decreased in most of the SRS grid cells from the RI/FS to the PDI/BL data. However, the percentage of RAL exceedance grid cells that decreased were lower than the sitewide percentage for total PCBs, and the percentage of RAL exceedance cells that increased for total DDX were greater than those that decreased. This suggests that there is less natural recovery occurring in the areas with RAL exceedances, and concentrations may even be increasing for total DDX. Dioxin/furan concentrations all increased relative to the RI/FS data, with 2,3,7,8-TCDD showing an increase in concentrations in both the sitewide and RAL exceedance grid cells.

Generally, the PDI/BL surface sediment data are consistent with the site CSM. The CSM relies on natural recovery occurring throughout approximately 84% of the site, with decreases in sediment concentrations due to upland source control and clean sediment deposition. Some area and COC combinations reflect those declines. Other areas that are less prone to natural recovery do not appear to have declined. As a decision framework, the ROD was designed to incorporate new data and accommodate the changes that have taken place over the 10 to 20 years since the RI/FS data were collected. During the design process, the PDI/BL surface and subsurface sediment data will be used in conjunction with higher resolution design samples to refine SMA footprints and determine further actions.

### **1.1.3. Equivalence (How do the current site concentrations compare to the background Upriver Reach?)**

To assess current surface sediment conditions in a statistically sound way, surface area weighted average concentrations (SWACs) were calculated. By area weighting and only using the unbiased PDI/BL data, an unbiased estimate of the average concentrations can be calculated. These SWACs can be compared against the CULs to determine how close or far average concentrations are from achieving the remedy goals. An added benefit to only using the unbiased PDI/BL data in the SWAC calculations is the ability to assess the uncertainty in these SWAC estimates. Uncertainty is represented by 95% confidence intervals around the SWAC.

**Figures 1-18a through 1-18j** show the SWACs and their error bars for the ROD Table 21 COCs. Each figure presents a different COC and shows the sediment CUL for that respective COC. For total PAHs, the CUL is higher than the SWAC; this CUL is based on ecological risk to benthic organisms. However, the SWAC for cPAHs, a subset of total PAHs, has a much lower CUL owing to their cancer risk in humans.

For all the ROD Table 21 COCs, the surface sediment SWACs are higher in the site than in the background Upriver Reach. **Figures 1-18a, 1-18d, and 1-18e** show wide 95% confidence intervals in the Upriver Reach that overlap the confidence intervals of the site for total PCBs; 2,3,7,8-TCDD; and 1,2,3,7,8-PeCDD, respectively. These are likely due to a limited number of PDI/BL samples collected in the Upriver Reach (n = 30) and a wide range of concentrations observed. Additionally, the overlap of the 95% confidence intervals between the site and the Upriver Reach indicate that

there are not enough Upriver Reach samples to statistically determine differences in the SWACs. Further sampling of the site and Upriver Reach is needed to confirm that CULs are being met. As additional sampling is performed, the wide-ranging confidence intervals will get smaller and allow for conclusive determinations of statistical difference.

Additionally, to evaluate remedial effectiveness before, during, and after the cleanup, sediment concentrations within the site are compared to those in the background Upriver Reach. This comparison is called equivalence testing and is based on the reverse null hypothesis as described in *Revised Working Draft Sampling Plan for Pre-Remedial Design, Baseline, and Long-Term Monitoring* (EPA 2017b). Equivalence testing is performed by calculating the ratio of the site and reference area geometric means. When the 95% upper confidence limit (UCL) for the ratio is less than an established equivalence factor, the site and reference area are deemed to be statistically equivalent and the remedy is working as planned. To assess current equivalence, a proposed equivalence factor of 1.5 was selected and the ratio between the site and Upriver Reach, site and Downtown Reach, and Downtown Reach and Upriver Reach geometric means were compared against this factor (**Figures 1-19a through 1-19c**). The ratio of 1.5 for determining equivalence allows for uncertainty in the data and may be adjusted based on future statistical evaluations. Determination of whether the site has reached equivalence with the Upriver Reach requires a series of repeated sampling events conducted as part of the long-term monitoring program.

The equivalence testing shows that naphthalene is the only ROD Table 21 COC with site concentrations currently equivalent to either the Downtown Reach or the Upriver Reach. The Downtown Reach is more similar to the site than the Upriver Reach, with the highest ratio at approximately 2.5 for cPAHs (site to Downtown Reach) and 4 for total PAHs (site to Upriver Reach). The Downtown Reach is approaching equivalence with the Upriver Reach at an equivalence factor of 1.5, suggesting that ongoing cleanup and source control in the Downtown Reach will provide for more recovery of sediment contamination.

## **1.2. Sediment Management Area Delineation**

The PDI/BL data are insufficient on their own or combined with the RI/FS data for full delineation of the SMAs, which has also been acknowledged by the Pre-RD Group in their *PDI Evaluation Report* (AECOM and Geosyntec 2019a). Higher spatial density samples will be required during remedial design to delineate the SMAs in both the surface and subsurface sediment. However, the PDI/BL data are useful for understanding the current lateral extent of the surface sediment SMAs at an FS level (+50%/–30% cost accuracy).

The lateral extent of the surface sediment SMAs was determined with the PDI/BL data following the methodology described in FS Appendix C (EPA 2016b) and was then compared to the lateral extent of the ROD SMAs. Both the unbiased SRS and biased SMA samples in the PDI/BL data were included in the SMA calculation. **Figure 1-20** shows the overlap between the PDI/BL and ROD SMAs with estimated areas of 360 and 365 acres, respectively. The similarity in the lateral extent of the PDI/BL and ROD SMA footprint suggests that the contaminated areas of the site have not changed substantially and that the hot spots are still present and resistant to natural recovery. There are differences between the two footprints, likely due to decreases in some COCs (e.g., total PCBs) and increases in others (dioxins/furans). There are also different sample counts for different COCs between the RI/FS and PDI/BL data, leading to differences in the interpolated SMAs. For example,

the PDI/BL data had 655 samples each for the six focused COCs while the RI/FS data had between 1,100 and 1,600 samples for total PCBs, total PAHs, and total DDx but only 220 samples for the dioxins/furans. **Figures 1-21a** through **1-21e** show the technology assignments for the PDI/BL SMAs following the ROD technology application decision tree ([TADT]; ROD Appendix I Figure 28 [EPA 2017a]).

The PDI/BL and RI/FS data were then combined into a single set and used to calculate the surface sediment SMA footprints. These combined data estimate an overall SMA footprint of 375 acres, which is not substantially different from the ROD (365 acres) and PDI/BL (360 acres) SMA footprints. These different estimations of SMA footprints are likely within the margin of error as discussed in FS Appendix I (EPA 2016b). **Figure 1-22** shows the technology assignments for the combined PDI/BL and RI/FS SMA footprints following the ROD TADT.

### **1.3. Conclusions**

The various spatial and statistical analyses performed with the PDI/BL surface sediment data suggest that there are not substantial changes in COC concentrations since the RI/FS data were collected 10 to 20 years ago. The concentration decreases observed in the PDI/BL data are predominantly associated with the areas selected in the ROD for monitored natural recovery (MNR), and the more contaminated hot spot areas still contain RAL exceedances. It is necessary to proceed with remedial design and remedial action to begin the cleanup of the site.

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## **2. Subsurface Sediment RAL Exceedances in Areas of Dioxin/Furan Surface Contamination**

Dioxin/furan congeners are five of the 11 COCs that drive the areas of active sediment remediation in the site. These COCs, listed on ROD Table 21, are 2,3,7,8-TCDD; 1,2,3,7,8-PeCDD; 2,3,4,7,8-PeCDF; 2,3,7,8-TCDF; and 1,2,3,4,7,8-HxCDF.

2,3,7,8-TCDD; 1,2,3,7,8-PeCDD; and 2,3,4,7,8-PeCDF are sediment focused COCs that have RALs, and 2,3,7,8-TCDF and 1,2,3,4,7,8-HxCDF have PTW concentration thresholds owing to their widespread distribution and toxicity. Dioxins/furans are carcinogenic and bioaccumulative and pose significant human health and ecological risks at very low concentrations (EPA 2016a). Additionally, concentrations in sediment and biota are generally low and stretch the limits of the best available laboratory methods, leading to difficulties in measurement and interpretation of results.

### **2.1. Development of Site Dioxin/Furan CULs and RALs**

The RI/FS surface sediment samples analyzed for dioxins/furans are in targeted areas of the site and have less spatial coverage than the other COCs. However, additional samples were collected in the Upriver Reach to define background concentrations. To adequately address dioxin/furan contamination at the site, the ROD established individual sediment CULs and fish tissue target levels for 2,3,7,8-TCDD; 1,2,3,7,8-PeCDD; 2,3,4,7,8-PeCDF; 2,3,7,8-TCDF; and 1,2,3,4,7,8-HxCDF. These five dioxin/furan congeners contribute greater than 85% of the estimated cancer risk and non-cancer hazard associated with fish consumption (EPA 2017a, 2016b). The target levels for fish tissue are risk-based, whereas the sediment CULs are based on concentrations in the background reference area (i.e., Upriver Reach). The use of background concentrations for CULs is described in Section 9.1.4 of the ROD, “If background concentrations are higher than the cleanup level, EPA defaults to the background concentration as a matter of policy” (EPA 2017a).

The background concentrations for dioxins/furans were developed using the RI/FS sediment data collected in the Upriver Reach. These background RI/FS dioxin/furan data have frequencies of detection much less than 50% for four of the five dioxin/furan congeners of interest, making it inappropriate to calculate UCLs using only the detected results. Instead, the background concentrations, hence the ROD Table 17 CULs, were established as the 95% UCL of the method detection limits (corrected upwards for the organic carbon content). The sediment CUL for 1,2,3,4,7,8-HxCDF was calculated using detected results from the Upriver Reach.

The RALs for the focused COCs and PTW thresholds selected in the ROD are concentrations above which dredging and/or capping needs to occur to effectively reduce risks in a reasonable time frame. The PTW thresholds for dioxins/furans are risk-based concentrations (1 in 1,000 cancer risk), whereas the RALs are based on the relationship between achieving CULs, post-construction SWACs, and the number of acres remediated. Since the dioxin/furan CULs are based on background concentrations, the RALs for 2,3,7,8-TCDD; 1,2,3,7,8-PeCDD; and 2,3,4,7,8-PeCDF are also tied to background.

## 2.2. Subsurface Sediment RAL Exceedances in Areas of Dioxin/Furan Surface Contamination

The PDI/BL data provide the first sitewide spatial coverage of dioxins/furans in the surface sediment. These new data have allowed for additional FS-level delineation of SMAs following the method described in FS Appendix C (EPA 2016b). Additionally, this method can be used to approximate the contribution of dioxins/furans to the lateral extent of the surface sediment SMA.

**Figure 2-1** shows that the PDI/BL SMA footprints are approximately 360 acres, of which approximately 90 acres are due to dioxins/furans alone (orange hashed areas). **Figure 2-2** shows that the interpolated SMA containing both the RI/FS and PDI/BL data is approximately 375 acres, with 71 acres being due to dioxins/furans alone. However, within these 71 acres, nearly all the areas of dioxin/furan surface contamination are comingled with or adjacent to subsurface RAL exceedances for total PCBs, total PAHs, or total DDx. These subsurface RAL exceedances indicate that despite the presence of only dioxin/furan contamination at the surface, contamination from the other focused COCs remain at depth and need to be further investigated during remedial design.

## 2.3. Equivalence between the Site and Upriver Reach

Despite the current limited temporal dataset, an equivalence evaluation was performed using only the unbiased PDI/BL surface sediment samples (i.e., SRS only), allowing for an unbiased estimate of equivalence. See Section 1.1.3 for a description of equivalence testing. This was done to evaluate if the PDI/BL data are showing concentrations within the site that are approaching equivalence with background as a result of sediment deposition since the RI/FS data were collected. Additionally, as the PDI/BL data have higher frequencies of detection for the Upriver Reach dioxins/furans (40 to 87%) compared to the RI/FS data (2 to 49%), the equivalence analysis with the PDI data is less sensitive to non-detect samples. **Figure 1-19a** shows the ratios of geometric means for the ROD Table 21 COCs between the site and Upriver Reach in surface sediment.

Equivalence testing with the PDI/BL data show that none of the Table 21 COCs have achieved equivalence between the site and the Upriver Reach (at an equivalence factor of 1.5) except for naphthalene, which is not a significant driver of the cleanup. However, 2,3,7,8-TCDD and 1,2,3,7,8-PeCDD appear to be approaching equivalence. Although the ratio of the geometric mean for 1,2,3,7,8-PeCDD is below 1.5, the 95% UCL is above and therefore within the margin of error. Additionally, these two COCs have the lowest RALs/PTW thresholds for the five dioxins/furans owing to their high toxicity.

While the frequencies of detection for the Upriver Reach PDI/BL samples are higher than those from the RI/FS, there are more rigorous data handling and analysis procedures for the non-detects that may need to be explored. However, additional handling and analysis for the non-detects may only marginally increase the current COC equivalence ratios. Therefore, because the COC equivalence ratios are currently at or above the 1.5 equivalence level, additional analysis would not change the conclusion that dioxin/furan concentrations in the site are not equivalent to the Upriver Reach.

## 2.4. Conclusions

While the background-derived CULs and RALs represent very low concentrations, estimated dioxin/furan surface SMAs in the site are comingled with or adjacent to subsurface RAL

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exceedances for total PCBs, total PAHs, and total DDX. These areas need to be delineated with surface and subsurface sediment samples during remedial design. Additionally, equivalence testing shows that dioxin/furan concentrations in the site are higher than those in the Upriver Reach. It is necessary to proceed with remedial design and remedial action to begin the cleanup of the site.

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### 3. Deposition in SMAs and Updated RAL Curves

The site is a 10-mile reach of the lower Willamette River, which is a dynamic river system that experiences episodic deposition and erosion over a range of spatial and temporal scales. This can be seen in **Figures 3-1** and **3-2**, which show the absolute bathymetric change between the 2003 and 2009 bathymetry surveys and the 2009 and 2018 bathymetry surveys. Although the site is net depositional, sediment is not deposited uniformly, and some areas are net erosional or in dynamic equilibrium and subject to periods of oscillating deposition and erosion. It is therefore important to consider the patterns of deposition and erosion over long time scales. The ROD SMAs receive less sediment deposition than the areas outside of the SMAs (i.e., MNR areas) and are more erosional or dynamic. Combined with the high concentrations of the focused COCs, the ROD SMAs are resistant to natural recovery and require active remediation. Results from the 2018 bathymetry survey and surface sediment sampling are evaluated for their impact on the SMAs and RALs defined in the ROD.

#### 3.1. Deposition in SMAs

Bathymetry surveys during the RI were completed in 2002, 2003, 2004, and 2009, with the most recent survey completed in 2018 as part of the PDI/BL. These surveys are conducted to measure the sediment bed elevations. By comparing the various surveys over time, a picture of long-term river dynamics can be developed that shows whether an area is consistently depositional, consistently erosional, consistently neutral, or in dynamic equilibrium. The consistency of sediment deposition in the site was evaluated with the 2002, 2003, 2004, 2009, and 2018 bathymetry surveys as described in FS Appendix D (EPA 2016b). **Figures 3-3** and **3-4** show the percentage distribution of these categories in the ROD SMA footprints and the remainder of the site area (i.e., MNR areas outside of the active remediation area), respectively. The spatial distribution of the consistently depositional, consistently erosional, consistently neutral, and dynamic equilibrium areas is shown in **Figures 3-5a** through **3-5f**.

The bathymetric change analysis suggests that the ROD SMA areas are 30% consistently depositional, with 70% that are erosional, neutral, or in dynamic equilibrium. Coupled with the high concentrations of focused COCs in these areas, MNR will not successfully reduce contaminant concentrations in a reasonable time frame. The results of the consistency evaluation, including the 2018 bathymetry survey data, agree with the natural recovery evaluation performed during the FS and summarized in FS Appendix D (EPA 2016b). The areas outside of the ROD SMAs (i.e., remaining site area) where MNR is the selected technology are 56% consistently depositional, with 44% that are erosional, neutral, or in dynamic equilibrium. Additionally, these areas do not have focused COC concentrations greater than RALs, and therefore MNR should be successful.

The 2004 and 2018 bathymetric surveys were also directly compared to understand the absolute change in sediment bed elevation during this time. The 2004 and 2018 surveys occurred just before comprehensive surface sediment sampling events during the RI and PDI/BL, respectively. Therefore, these two surveys represent two appropriate points for direct comparison of sediment deposition and surface sediment chemistry. The results suggest that sediment deposition is not evenly distributed throughout the ROD SMAs. **Figure 3-6** shows the amount of net deposition in cubic yards that the SMAs (segregated by EPA proposed remedial design areas) received and the average thickness of this deposited sediment. **Figures 3-7a** through **3-7f** show the spatial distribution of deposition and erosion from 2004 to 2018.

Sediment deposition varies between areas in the ROD SMAs and ranges from –10,000 cubic yards (i.e., erosion) in the B4 area (river mile [RM] 11E) to 87,000 cubic yards in the B3 area (RM 9W). This results in average sediment thicknesses of the deposited sediment that range from –4.5 to 19 inches (about 1.5 feet). The zone of surface sediment is defined as the top 12 inches of the sediment bed, and only two proposed design areas (B3 and B8) received sediment deposition greater than surface depth thresholds over a 14-year period from 2004 to 2018.

### **3.2. Updated RAL Curves**

In the ROD, the surface sediment data from the RI/FS were used to develop RAL concentrations for the six focused COCs (EPA 2017a). The RAL concentrations consider the amount of material that would be addressed to achieve COC and risk reductions throughout the site. This is done with RAL curve plots, which compare the number of acres remediated against the post-remediation SWACs. The ROD selected higher RALs for the navigation channel (“B” RALs) compared to the remaining site area (“F” RALs) owing to the disconnected exposure pathways in the deeper navigation channel (EPA 2017a). Updated RAL curves for the focused COCs were developed with the PDI/BL surface sediment data to determine whether the relationship between concentration and area remediated has substantially changed since the RI/FS data were collected. **Figures 3-8a through 3-8f** show the RAL curves with data from the RI/FS (ROD Only; solid line), PDI/BL only (PDI Only; light dash line), and RI/FS and PDI/BL combined datasets (PDI Only; dark dash line).

The updated plots with the PDI/BL data show that the RAL concentrations for the focused COCs selected in the ROD<sup>1</sup> are still appropriate. The curves for total PAHs, total DDx, and 2,3,4,7,8-PeCDF indicate little change while those for 2,3,7,8-TCDD and 1,2,3,7,8-PeCDD show an increase in the area requiring active remediation. The area requiring active remediation for total PCBs appears to have decreased; however, the F RAL is still appropriate for substantial risk reduction in the nearshore areas without experiencing diminishing returns. The curves for the three datasets (RI/FS, PDI/BL, and RI/FS combined with PDI/BL) are generally similar but do contain differences. These differences are likely due to the 10 to 20 years between the RI/FS and PDI/BL data collection efforts and the more complete sitewide dioxin/furan sampling performed in 2018 during the PDI/BL.

### **3.3. Conclusions**

Seventy percent of the ROD SMAs are erosional, neutral, or in dynamic equilibrium compared with 44% of the remaining site area. Therefore, MNR will not sufficiently reduce risk in the SMAs in a reasonable time frame, and active remediation such as capping and dredging is necessary. The PDI/BL data indicate that the RALs selected in the ROD are still appropriate and, for natural recovery to be effective, these hot spot areas need to be remediated.

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<sup>1</sup> The proposed ESD proposes increasing the sitewide RAL for total PAHs from 13,000 to 30,000 micrograms per kilogram (µg/kg) based on updated risk and toxicity information in the EPA Integrated Risk Information System.

## 4. Upstream Sediment Trap Analysis and Willamette River Hydrodynamics

Upstream sediment trap deployments during the PDI/BL included four traps deployed across two transects, located at RMs 11.8 and 16.2 (shown with yellow circles in **Figure 4-1**). Each transect had a sediment trap deployed on the east and west side of the river. The PDI/BL sediment trap data assist in understanding the ROD Table 21 and Table 17 COC concentrations and qualitative spatial distribution of settleable suspended sediments upstream of the site.

### 4.1. PDI Sediment Trap Chemical Results

**Figures 4-2** and **4-3** summarize the PDI/BL sediment trap results for the ROD Table 21 COCs and Table 17 COCs (those not on Table 21), respectively, from the Round 1 (low stage), Round 2 (storm-induced), and Round 3 (high stage) deployments. Sediment trap COC concentrations were less than the ROD Table 17 CULs at both transects during all deployments for total PAHs, total DDx, the individual DDx isomers, metals (except for arsenic and mercury), aldrin, lindane, and tributyltin. Results for total PCBs were above CULs at RM 11.8 during the low stage deployment but were otherwise less than CULs. Most results for the dioxins/furans were above CULs at RMs 11.8 and 16.2 for all three sampling rounds. However, 75% of the results from the low-stage deployment were not detected due to elevated detection limits (greater than CULs) at the analytical lab. Therefore, these non-detect results for the dioxins/furans are shown at the lab-reported detection limits. Dioxin/furan results from the storm-induced flows deployment were all detected and generally had lower concentration than the low-stage deployment. Non-detects were present in the high stage deployment, but detection limits were less than CULs in all instances.

PDI sediment trap COC concentrations were also compared to the upstream sediment traps from the RI. **Figures 4-4** and **4-5** summarize the RI and PDI sediment trap data for the ROD Table 21 COCs and Table 17 COCs (those not on Table 21), respectively. Generally, the range of concentrations observed in the RI and PDI sediment traps is similar although dioxins/furans at the RM 16.2 transect appear to be higher in the PDI data. However, owing to the small sample sizes, differences in flow regimes during sediment trap deployments, sampling locations, and water depths, a robust statistical analysis is not possible. The RI and PDI sediment trap data are useful when comparing to CULs as individual samples and to observe qualitative differences between the different deployment flow regimes.

### 4.2. Hydrodynamics

Hydrodynamics play a role in the movement of suspended sediments. Flow reversals from tidal action cause water flow to oscillate upstream and downstream, potentially holding contaminated sediment within an area or even causing it to distribute upstream. The Willamette River experiences tidally induced flow reversals during low-water conditions, which typically occur during late summer and fall. The Round 1 deployment occurred during low river stage, with lower average discharge and more frequent flow reversals, whereas Round 2 occurred during the period of winter storms, with fewer flow reversals and higher average discharge. The Round 3 deployment was during the period of higher sustained flows along with winter storms, and flow reversals were rarely observed. **Table 4-1** summarizes the PDI sediment trap deployment periods and the associated hydrodynamic conditions. **Figures 4-6a** through **4-6c** show the Willamette River

discharge measured at the Morrison Bridge in downtown Portland (RM 13; U.S. Geological Survey [USGS] gage 1421170) throughout the three sediment trap deployment periods.

**Figures 4-6a** through **4-6c** indicate that the low stage deployment was subject to more frequent and higher magnitude flow reversals than the storm-induced flows and high stage deployments. Also, the RM 11E project area extends close to RM 11.8, has numerous ship berths, and experiences its highest levels of ship and barge traffic during summer and early fall when the Willamette River flows are lowest. This area has elevated PCB concentrations greater than RALs in surface sediment and a high potential for resuspension owing to tugs and propwash. Therefore, it is possible that the elevated PCB concentrations in the sediment traps deployed at RM 11.8 during the low-stage deployment result from local contamination. Further evidence for this is seen in the PCB concentration results at RM 11.8, which decreased during the Round 2 and Round 3 deployments, likely due to less shipping activity and higher flows, reflecting more predominant upstream background PCB concentrations rather than a backwash influence from local elevated source contamination.

River turbidity increased in Round 2 and Round 3 with increased discharge and runoff from winter storms. The increased turbidity does not appear to have an overall impact on the sediment trap COCs' chemistry as the Round 3 concentrations were predominantly lower than both the Rounds 1 and 2 results. **Figure 4-7** shows the relationship between discharge and turbidity for the three PDI sediment trap deployments.

### **4.3. Mass Loading Estimates**

Sediment and contaminant mass loading estimates were calculated with both the RI and PDI sediment trap data. These mass loading calculations are approximate and should only be used for relative, semi-quantitative calculations between sampling rounds. Because of uncertainty regarding the capture of suspended sediments by the sediment trap, the sediment trap data are not suitable for developing quantitative sediment loading estimates. The loading measurement represents the net sediment accumulation in each sediment trap over the duration of the deployment.

The sediment mass loading was calculated from the measured sediment volume, assuming an average sediment specific gravity of 2.65. Measurements of sediment density were not performed as part of this study. The calculated sediment mass loading for each PDI sediment trap is shown in **Figure 4-8**. The highest sediment loading during the PDI sampling occurred during the Round 3 deployment owing to the higher flows and greater volume of sediment that accumulated in the sediment traps. The constituent mass loading for the ROD Table 21 COCs and Table 17 COCs (those not on Table 21) was calculated based on the calculated sediment loading. The mass loading for each ROD Table 21 and ROD Table 17 COC is presented in **Figures 4-9** and **4-10**, respectively.

Sediment and contaminant mass loading were also estimated for the upstream RI sediment traps, and the results were compared to the PDI/BL data. The range of estimated sediment mass loading rates for the RI was similar to the PDI; however, higher flows and therefore higher sediment loading occurred during the RI sampling (**Figure 4-11**). This resulted in similar contaminant mass loading estimates between the two studies (**Figures 4-12** and **4-13**).

#### **4.4. Conclusions**

Flow reversals during the low-stage deployment suggest that total PCB concentrations at RM 11.8 are possibly due to local contamination rather than an upstream source. The three dioxins/furans were predominantly above CULs during all three sampling rounds; however, the large percentage of non-detect samples during the low-stage deployment make it difficult to draw any meaningful conclusions from these results. The PDI sediment trap data suggest that upstream suspended sediments are relatively clean and do not represent an uncontrolled source of contamination to the site.

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## 5. Subsurface Modeling Evaluation and SMA Delineation

The ROD SMAs were developed using chemical data only from the surface sediment (top 0 to 30 centimeters [cm]) owing to the limited number of subsurface sediment cores collected during the RI/FS (EPA 2016b, 2017a). While the RI/FS subsurface sediment core data are limited in significant areas of the site for completing remedial design, additional sediment data in both the surface and subsurface have been collected since the RI/FS database was finalized. These data, in addition to the RI/FS cores, have been incorporated into a sitewide subsurface sediment model. This new model will be used to support many components of remedial design decisions, including:

- To identify areas where higher density subsurface sediment core sampling is needed to horizontally and vertically bound areas of contamination
- To estimate volumes of contaminated sediments in SMAs to generate more accurate disposal costs
- To explore how SMAs change over time with new bathymetry surveys and additional sediment samples

This model will be continually updated as new subsurface data are collected throughout the site during remedial design.

### 5.1. Subsurface Sediment Model Development

The subsurface sediment model has been developed using a three-dimensional (3-D) geological modeling software program called Leapfrog Works (v2.2.2) developed by Seequent. Leapfrog Works uses sample locations, chemistry data, and mathematical interpolations to develop 3-D estimates of areas of interest.

For Portland Harbor, sediment samples from the site RI/FS database, RM 11E supplemental RI/FS database (GSI Water Solutions, Inc. [GSI] 2014), RMs 5 to 6 sediment sampling database (NewFields 2016), and PDI/BL database have currently been incorporated into the 3-D model. As new remedial design samples are collected, the model will be updated to include this new information.

Sediment sample locations throughout the site were draped over the 2018 bathymetric surface (i.e., river bottom elevations) to accurately place them in 3-D space. From there, concentrations of the ROD Table 21 COCs were evaluated to determine if they were greater than the applicable RAL and/or PTW threshold. A 3-D field of interpolated sediment concentrations for the Table 21 COCs was developed, and the union of the individual COC exceedances of RALs and/or PTW thresholds were mapped as subsurface SMAs. **Figures 5-1a** through **5-1f** show the lateral extent (i.e., two dimensions) of the modeled subsurface SMAs compared to the surface sediment only ROD SMAs.

**Figures 5-1a** through **5-1f** show that the majority of the modeled subsurface SMA footprints are encompassed by the ROD surface-only SMA footprints. However, there are some areas where subsurface contamination is estimated to be present where the surface (top 0 to 30 cm) may not contain RAL or PTW exceedances. This is consistent with the CSM in the ROD where clean sediment may be depositing in areas of subsurface contamination. These estimated areas of subsurface contamination should be further evaluated in remedial design.

## **5.2. Current Applications**

The 3-D sediment model has currently been used to identify subsurface sediment RAL exceedances outside of the Pre-RD Group's preliminary refined PDI surface-only SMA footprint presented in their *Pre-Remedial Design Footprint Report* (AECOM and Geosyntec 2019b). This analysis was conducted throughout the site, and **Figures 5-2** through **5-12** show how the modeled subsurface SMAs were compared against and differed from the Pre-RD Group's surface-only SMAs in a given area of the site. Multiple instances of insufficiently bounded subsurface sediment cores with RAL exceedances outside of the Pre-RD Group's SMAs were identified. These areas need to be explored during remedial design, and the subsurface modeling is helping to identify where data gaps exist.

Additionally, the 3-D sediment model was used to identify the presence of total PCBs, total PAHs, and/or total DDx in subsurface sediments in areas of dioxin/furan contamination in surface sediment as described in Section 2. This analysis determined that dioxin/furan RAL exceedances in the surface sediments are generally collocated with the other focused COCs (PCBs, PAHs, and/or DDx) in the subsurface (**Figure 2-2**).

## **5.3. Conclusions**

The 3-D sediment model is a useful tool that can be used during remedial design to better understand the areas of contamination, estimate volumes (and therefore cost) of contaminated sediment disposal, and truth-check design plans on smaller spatial scales. The model will be continually updated with new data as they are collected and will evolve and inform throughout the remedial design process.

## 6. Surface Water Evaluation

The PDI surface water sampling consisted of three sampling rounds at seven transects located within the site, entrance to Multnomah Channel (MC), and the D/U Reach. The transects were located at RMs 1.9, 3 (mouth of MC), 4, 7, 8.8, 11.8, and 16.2 and are shown in **Figure 6-1**. Samples were collected along the east shore, west shore, and in the navigation channel and were composited into a single sample at each transect. The PDI surface water results assist in understanding the COC concentrations and estimated spatial distribution of COC mass loading to, within, and leaving the site.

### 6.1. PDI Surface Water Chemistry Results

The PDI surface water sampling targeted three different flow conditions: low river stage, stormwater flows, and high river stage. The three sampling rounds are summarized in **Table 6-1**. **Figure 6-2** shows the Willamette River discharge from August 2018 through February 2019 when sampling occurred.

The surface water samples were collected as particulate and filtered samples and analyzed for the ROD Table 17 surface water COCs. Results for the total fraction were generated by summing the results from the particulate and dissolved (i.e., filtered) fractions. **Figures 6-3a** through **6-3e** show the surface water results from the seven transects for the ROD Table 17 COCs (error bars represent the range of concentrations measured during the three sampling rounds). The PDI average concentrations show that PCBs, cPAHs, and total dioxins/furans expressed as 2,3,7,8-TCDD toxic equivalent (TEQ) were above the surface water CULs at all transects. These CULs were selected in the ROD based on existing regulatory requirements for human health criteria (U.S. Clean Water Act; Oregon Water Pollution Act) and represent very low concentrations.

Additionally, the highest COC concentrations measured were generally located immediately downstream of areas of known sediment contamination. For example, PCBs were highest at Transects 3 and 5, which are downstream of PCB sediment contamination at Terminal 4/Schnitzer Steel (RM 4) and Gunderson (RM 9), respectively. This pattern was consistent for cPAHs (Transect 3; Gasco and other oil companies), DDx (Transect 4; Arkema), and total dioxins/furans (Transects 3 and 4; Terminal 4 and Arkema, respectively). While the concentrations of DDx were below its risk-based CULs at all transects, the DDx isomers dichlorodiphenyltrichloroethane (DDT), dichlorodiphenyldichloroethene (DDE), and dichlorodiphenyldichloroethane (DDD) have much lower regulatory-based CULs than DDx and were above their respective CULs in the site (**Figure 6-3b**).

Average PDI surface water concentrations were also compared between the different reaches for the different sampling rounds for PCBs, cPAHs, DDx, and 2,3,7,8-TCDD TEQ. **Figures 6-4a** and **6-4b** show the average concentrations for the site (Transects 1 through 5) and upstream D/U Reach (Transects 6 and 7). Concentrations decreased for PCBs and cPAHs as flow increased, likely owing to dilution in the water column, whereas DDx concentrations were highest during Round 3, and 2,3,7,8-TCDD TEQ average concentrations were consistent across sampling rounds. **Figures 6-4a** and **6-4b** show that roughly 50% of the contaminant mass is bound to suspended sediments, and 50% is dissolved in the water column for PCBs and DDx while nearly all the mass is from suspended sediments for cPAHs and total dioxins/furans. All the COCs do not readily dissolve in water and instead bind strongly to sediment particles where they may be transported downstream or settle to

the river bottom and accumulate in the tissues of fish and other aquatic organisms (Agency for Toxic Substances and Disease Registry [ATSDR] 2019).

Aggregated averages were also calculated to qualitatively compare concentrations between the site and D/U Reach across sampling rounds (**Figure 6-5**). Concentrations for detected results were generally higher in the site than in the D/U Reach.

## **6.2. Willamette River Hydrodynamics**

Hydrodynamics play a role in the movement of contaminants in both the particulate (i.e., suspended sediments) and dissolved phases. A detailed discussion of hydrodynamics as they pertain to the PDI sediment trap results is included in Section 4.

Despite the threefold increase in average discharge from the Round 1 to the Round 2 surface water sampling events, the turbidity measured at each transect was slightly lower during Round 2. This could possibly be due to brown algae present in the Willamette River, which blooms during the summer and can be detected by optical turbidimeters like the one used during the PDI surface water sampling. Frequent filter changes and higher chlorophyll measurements in Round 1 versus Round 2 provide additional lines of evidence for the impact of algae on the measured turbidity. Additionally, the storm that was targeted in Round 2 was a low magnitude storm event (**Figure 6-2**) that would not increase turbidity as dramatically as a larger storm or sustained higher flows as occurred during the Round 3 sampling. The similarities in turbidity between the Rounds 1 and 2 sampling along with the higher flows during Round 2 could have had a dilution effect on the COC mass, leading to the lower concentrations observed.

### **6.2.1. Total Suspended Solids Mass Loading**

At the USGS gage 14211720, Willamette River at Portland, Oregon, the USGS collects periodic total suspended solids (TSS) measurements. During the 2018/2019 PDI sampling program, TSS measurements were collected approximately monthly. However, the USGS also collects continuous instantaneous turbidity measurements at this gage. To generate a continuous timeseries of estimated TSS concentrations at the USGS gage, a linear regression was developed between median daily turbidity and measured TSS from 2009 through 2019 (**Figure 6-6**). Measured TSS concentrations from Transect 6 in the PDI sampling program were also included in this regression. A stronger relationship between measured and modeled TSS concentrations was achieved by omitting the two highest TSS concentrations from the regression, which appear to be outliers. These are identified as open triangles in **Figure 6-6**. The overall regression is statistically significant ( $R^2 = 0.90$ ;  $p < 0.01$ ).

To evaluate the strength of the linear regression, the measured TSS concentrations were compared with modeled TSS concentrations (**Figure 6-7**). Most of the measurements fall close to the 1 to 1 line, suggesting that the linear regression model is a good predictor of measured TSS below 50 milligrams per liter (mg/L). Limited high TSS data exist to validate the model, but available data suggest that the model is biased low relative to measured data for high TSS concentrations ( $> 50$  mg/L). The model could be improved by adding additional TSS samples collected during high flow conditions.

The linear regression was applied to the timeseries of median daily turbidity measurements to generate a predicted TSS concentration and mass loading timeseries. The predicted mass loading

timeseries, compared to the calculated TSS mass loading based on the monthly USGS TSS samples, is presented in **Figure 6-8**. The predicted TSS loading is biased low during low flow but is representative of measured TSS loading during higher flow conditions.

### **6.3. Contaminant Mass Loading**

The mass loading at each transect and within the site and D/U Reach were calculated based on the daily average tidally filtered discharge reported at USGS gage 14211720 (Willamette River at Portland, Oregon). Because the increase in drainage area between the upstream transect and the downstream transect is small relative to the total drainage area of the Willamette River, no adjustments were made to the discharge through the study area upstream of the MC.

Special consideration was given to loading calculations at Transects 1 and 2 to account for the MC distributary. Continuous flow gaging is not available on the MC or downstream of the distributary. To understand the relative distribution of flow between the MC and the mainstem Willamette River, the EPA ran the CE-QUAL-W2 model of the Columbia River/Willamette River system over the 4-year period between January 1999 and December 2002. (EPA 2016a). This model found that the distribution of flow between the MC and the Willamette River depends on the tide, the flow in the Willamette River, and the flow in the Columbia River. Over the 4-year model period, the average flow distribution was 60% to the MC and 40% to the Willamette River. This flow distribution was used to calculate the mass loading at Transects 1 and 2.

Longitudinal profiles of mass loading at each transect are presented in **Figures 6-9a** through **6-9e** (error bars represent the range of calculated COC mass loading estimates from the three sampling rounds). The lines represent the mean mass loading across the three sampling events, and the error bars represent the range of observations. **Figures 6-10a** and **6-10b** show the average PDI mass loading rates for PCBs, cPAHs, DDx, and 2,3,7,8-TCDD TEQ for each sampling round. The surface water mass loading entering, within, and leaving the site increased with increasing discharge. Additionally, the loading rates were generally higher within and leaving the site than for the COC mass entering the site, suggesting that there is more contamination present within the site than there is upstream. This is consistent with the ROD CSM (EPA 2017a) and the PDI/BL surface sediment data summarized in Section 1. Aggregated averages were also calculated to qualitatively compare COC mass loading estimates between the site and D/U Reach across sampling rounds (**Figure 6-11**). Surface water COC mass loading was also visualized for PCBs, cPAHs, DDx, and 2,3,7,8-TCDD TEQ with a series of maps (**Figures 6-12, 6-13, 6-14, and 6-15, respectively**).

#### **6.3.1. Dioxins/Furans**

The D/U Reach mass loading rates for 2,3,7,8-TCDD TEQ increased with increasing discharge similar to the other COCs in the site and upstream (**Figure 6-10a**). This, along with the concentration results shown in **Figure 6-3c**, suggests that surface water with dioxin/furan concentrations greater than the CUL are flowing into the site from upstream, with greater concentrations and loading rates at higher flows.

### **6.4. PDI and RI Surface Water Comparison**

Historical concentrations for the ROD Table 17 COCs, collected as part of the RI between 2004 and 2007, were compared against the 2018/2019 PDI surface water sampling. RI samples included both those collected with a peristaltic pump and with a high-volume sampler for ultra-low detection

limits. RI surface water samples were selected for comparison based on their proximity to a PDI sample, and preference was given to the high-volume sampler method when results for both methods existed. Water quality metrics such as turbidity, TSS, and concentrations of COCs are often impacted by hydrologic conditions, including flow rate and relative hydrograph position at the time of sample collection (i.e., rising limb, peak, or falling limb). The comparisons in this report between the RI and PDI surface water samples lack hydrologic context to support evaluation of these data and are therefore qualitative.

This evaluation in general found reductions in organic concentrations between the RI and PDI water quality sampling events but little change in metals concentrations. A comparison of the range of RI and PDI surface water results is presented for each ROD Table 17 COC by river mile in **Figures 6-16a** through **6-16e**. In these figures, the range of results for the RI sampling is shown as a box and whisker plot. The mean PDI sampling is shown as a continuous line, and the shaded area represents the range of concentrations across the three sampling rounds. The total fraction is shown in blue and the dissolved fraction in orange. The y-axis scale is logarithmic to more clearly show the range of RI and PDI surface water concentrations. Aggregated averages were also calculated to qualitatively compare COC concentrations between the site and D/U Reach for the RI and PDI (**Figures 6-17a** and **6-17b**).

## **6.5. Conclusions**

PDI surface water sampling results were above the very low, regulatory-based CULs for PCBs, cPAHs, and total dioxins/furans both in the site and upstream. Concentrations in surface water and mass loading rates were highest near known hot spot areas of sediment contamination, suggesting that remediation of these areas and ongoing source control will help in achieving CULs. Future repeated rounds of surface water sampling during and after construction of the remedy will allow for estimating long-term trends in COC concentrations and mass loading rates.

## 7. Fish Tissue Statistical Evaluations

At Portland Harbor, different species of fish and shellfish are important food sources for subsistence, tribal, and recreational fishers. Both non-resident (e.g., salmon) and resident (e.g., smallmouth bass) fish are caught in the site and consumed by people living in the Portland area. The consumption of resident fish presents the greatest exposure risk to people from the site COCs (EPA 2017a). Smallmouth bass specimens have been caught and analyzed for site COCs during multiple studies and can be evaluated over time. This report summarizes fish tissue results in smallmouth bass from the 2018 PDI and compares them to the RI data.

### 7.1. PDI Fish Tissue Chemistry Results

During the 2018 PDI, smallmouth bass specimens were collected in the site (n = 95), Downtown Reach (n = 21), and Upriver Reach (n = 19) and were analyzed for the ROD Table 17 fish tissue COCs. Fish tissue concentrations in this report were assessed as and compared to ROD Table 17 target levels as whole body rather than fillet portions. **Figures 7-1** through **7-12** show the smallmouth bass capture locations and their COC concentrations for PCBs, DDx, dioxins/furans, aldrin, dieldrin, total chlordanes, bis(2-ethylhexyl)phthalate (BEHP)<sup>2</sup>, and mercury, respectively. Smallmouth bass specimens with the highest COC concentrations were generally collocated with elevated sediment concentrations. For example, smallmouth bass with the highest concentrations of total PCBs were collected from the SMAs of known PCB sediment contamination (Schnitzer Steel, Willamette Cove, Swan Island Lagoon, Gunderson, and RM 11E). This same pattern was consistent for DDx (Arkema) and dioxins/furans (Arkema, Willamette Cove, and Gunderson). Higher concentrations of these COCs in sediment lead to more bioaccumulation and higher levels in fish tissue.

Organochlorine compounds such as PCBs, DDx, and dioxins/furans are lipophilic and predominantly accumulate in the fatty tissues of fish. Their concentrations are often adjusted by normalizing the COC result to the amount of lipid present in the specimen to account for variation in tissue lipid content. However, this normalization is not always appropriate (Hebert and Keenleyside 1995). This topic is discussed in further detail in Section 10. The statistical distribution of fish lipid was evaluated for each of the fish tissue sampling study years of interest (2002, 2007, 2011, 2012, and 2018) and is shown in **Figure 7-13**. The overlap of the lipid distribution in the different years and reaches suggests that there are not statistical differences in lipids between the study years or reaches that would necessitate lipid normalization. This was further supported by a regression analysis discussed later in this section.

**Figures 7-14a** through **7-14l** show the arithmetic and geometric average concentrations for total PCBs, DDx, dioxins/furans, aldrin, dieldrin, total chlordanes, BEHP, and mercury, respectively, compared against the risk-based tissue target levels. These fish tissue COCs have very low tissue target levels owing to their carcinogenic and/or bioaccumulative properties. Average concentrations of total PCBs, DDx, and 2,3,4,7,8-PeCDF were higher in the site than in the Downtown Reach and Upriver Reach; average concentrations of 2,3,7,8-TCDD and 1,2,3,7,8-PeCDD

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<sup>2</sup> Of the 135 smallmouth bass specimens collected during the PDI, 117 had elevated detection limits for BEHP of 20,000 µg/kg. A meaningful comparison to the ROD Table 17 BEHP fish tissue target level cannot be made for these samples due to the elevated detection limits.

were highest in the Upriver Reach. However, the Upriver Reach average for 2,3,7,8-TCDD and 1,2,3,7,8-PeCDD is elevated relative to the site and Downtown Reach owing to a single fish specimen that had the highest measured concentrations of these COCs during the PDI. When this high concentration fish is excluded, Upriver Reach average concentrations for 2,3,7,8-TCDD and 1,2,3,7,8-PeCDD are slightly less than the site average. Currently, it is unclear why this single smallmouth bass specimen in the Upriver Reach had such high concentrations of 2,3,7,8-TCDD and 1,2,3,7,8-PeCDD. **Figures 7-15a through 7-15l** show the ratio of the PDI results compared to the ROD Table 17 fish tissue target level for the 12 COCs evaluated in this report.

Additionally, the distribution of concentrations for the fish tissue COCs was evaluated for differences between the reaches and the Pre-RD Group's proposed river segments for the site. The proposed river segments are approximately 2 to 3 river miles long and separated east and west by the center of the river (**Figure 7-16**). For the 12 bioaccumulative fish tissue COCs evaluated, the concentrations are generally higher in the site than the D/U Reach. However, there do not appear to be any substantial differences in the distributions between the proposed river segments within the site (**Figure 7-17a through 7-17l**).

### **7.1.2. Fish Lipid and COC Regression Analysis**

Fish tissue lipid percentages were evaluated against COC concentrations using a first order linear regression method to determine if there is a statistical relationship between lipid and COC concentrations. Regressions were performed for the 12 bioaccumulative fish tissue COCs of interest for the 2002, 2007, 2011, 2012, and 2018 study years when data were available. Additionally, this analysis was used to inform whether it is appropriate to lipid normalize the PDI fish tissue COC concentrations. Generally, there is not a statistical relationship between lipid and COC concentrations for the bioaccumulative COCs of interest in the different study years (**Figures 7-18a through 7-18l**). This suggests that it is not necessary to lipid normalize the fish tissue COC data at Portland Harbor.

## **7.2. Equivalence Testing**

To evaluate remedial effectiveness before, during, and after the cleanup, fish tissue concentrations within the site are compared to those in the Upriver Reach (i.e., background reference area). This comparison is called equivalence testing and is based on the ratio of the site and Upriver Reach (and the ratio of the Downtown Reach and Upriver Reach) geometric means. When the 95% UCL for the ratio is less than 1.5, the site (or Downtown Reach) and background reference area are deemed to be statistically equivalent and the remedy is achieving the intended goals.<sup>3</sup>

The equivalence testing was initially performed using the surface sediment data, which showed that PCBs, PAHs, DDx, and dioxins/furans in the site are not equivalent with the Upriver Reach (see Section 1.1.3). The results of the fish tissue equivalence testing are similar to those of the surface sediment and are summarized in **Figures 7-19a through 7-19g**. The equivalence testing suggests that the PDI smallmouth bass tissue concentrations for total PCBs, DDx, and dioxins/furans (except

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<sup>3</sup> The ratio of 1.5 for determining equivalence allows for uncertainty in the data and may be adjusted based on future statistical evaluations. Determination of whether the site has reached equivalence with the Upriver Reach requires a series of repeated sampling events conducted as part of the long-term monitoring program.

for 2,3,7,8-TCDD) in the site are not equivalent with those in the Upriver Reach and are therefore statistically higher in the site. The concentrations of 2,3,7,8-TCDD in the site appear to be equivalent with the Upriver Reach, but this may be driven by the anomalously high specimen collected in the Upriver Reach. Additional future sampling rounds are required to assess whether this trend is consistent over time. It is important to note that although 2,3,7,8-TCDD concentrations in the site are equivalent with the Upriver Reach, none of the samples were below the ROD Table 17 risk-based tissue target level.

### **7.3. Concentration Trend Analysis**

Smallmouth bass sampling occurred in the site and upstream in 2002, 2007, 2011<sup>4</sup>, 2012, and 2018. These sampling events varied in the locations from which fish specimens were collected, which COCs were analyzed, and whether a sample result was from a single fish or multiple fish combined. The 2002 and 2007 sampling events during the RI obtained fish tissue data for all the site COCs but had fewer results due to creating composite samples ( $n = 17$  for each study). The 2012 sampling event included more samples collected throughout the site and upstream ( $n = 92$ ) and no compositing, but specimens were only analyzed for total PCBs. Sampling during the 2018 PDI had a large, well-distributed sample size ( $n = 135$ ), and specimens were analyzed for all the ROD Table 17 COCs. **Table 7-1** summarizes the number of samples for each study year sitewide and within the Pre-RD Group's proposed river segments.

Despite the differences in the different sampling events, it is possible to evaluate how concentrations for the fish tissue COCs have changed from 2002 to 2018. A first order decay model was developed to estimate the average rates of change in units of per year ( $\text{yr}^{-1}$ ). The model assumes a common rate for the site but with differing absolute concentrations in different areas. Average rates of change for the site were estimated for total PCBs, DDx, and dioxins/furans, which are summarized in **Figure 7-20**.

The trend analysis suggests that sitewide fish tissue concentrations of PCBs; DDx; 2,3,7,8-TCDD; and 2,3,7,8-TCDF have decreased since 2002, albeit at rates of change less than  $10\% \text{ yr}^{-1}$  (i.e., first order rate coefficients greater than  $-0.1 \text{ yr}^{-1}$ ). This is consistent with the ROD, which states that concentrations will decrease over time due to active remediation and natural recovery (EPA 2017a). Because sitewide remedial action has not yet occurred, these small decreases shown in **Figure 7-20** are due to natural recovery and source control alone. With these rates of change less than  $10\%$  per year, natural recovery alone is insufficient to reduce concentrations substantially, and active sediment remediation is needed to achieve fish tissue target levels.

Additionally, as presented in the ROD, fish tissue concentrations in the site are assessed at smaller spatial scales of 0.5 to 1 river mile, separating the east and west sides of the river (EPA 2017a). These small spatial scales were developed as part of the risk assessments in the RI based on the home ranges of the resident smallmouth bass and the human health exposure scenarios (EPA 2016a). When evaluating sitewide, average concentrations are lower owing to relatively clean

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<sup>4</sup> An issue was encountered during sample processing at the lab for the 2011 study year such that only total PCBs and lipids were analyzed. Additionally, the 2011 study contained specimens typically larger than the size range targeted at Portland Harbor. For these reasons, the 2011 study year has not been included in the temporal change analysis.

samples being interspersed with those of higher concentration. At smaller spatial scales, the high concentrations carry more weight and the resulting averages are higher.

As discussed above, the 2002, 2007, and 2012 studies have limited spatial distribution of samples collected or a limited number of COCs analyzed, which limit the ability to understand long-term trends at the 0.5 to 1 river mile spatial scale. Future sampling events that repeat the 2018 PDI will be used to assess the rate of decrease of COCs in fish tissue as part of the long-term monitoring program presented in the ROD. However, with the data currently available, it is possible to estimate the rate of change at larger spatial scales than the risk-based ones in the ROD, such as the Pre-RD Group's proposed river segments. These spatial scales are not included in the ROD and have not been substantiated by the EPA.

**Figures 7-21a** through **7-21g** show the rates of change for PCBs, DDx, and dioxins/furans in the Pre-RD Group's proposed river segments. These results confirm that there is much more variability at smaller spatial scales in the existing data and that fish tissue COCs showing sitewide decreases since 2002 have not decreased in areas with collocated sediment contamination. For example, Segment 2W has elevated levels of PCBs, DDx, and dioxins/furans in sediment and shows no decrease in fish tissue concentrations for these COCs.

#### **7.4. Conclusions**

Average fish tissue concentrations from the 2018 PDI are above target levels in the site, Downtown Reach, and Upriver Reach for PCBs, DDx, and dioxins/furans. Additionally, the concentrations in the site are not equivalent with those in the Upriver Reach, and there has been only a limited decrease due to natural recovery since 2002. These analyses suggest that active sediment remediation in the site is required to achieve fish tissue target levels.

## 8. Background Porewater Evaluations

Porewater is the water between and surrounding grains of sediment in the bed of a water body. In tidally influenced water bodies like the lower Willamette River, it is usually a combination of groundwater and surface water owing to the interaction between the two zones. The PDI focused its porewater sampling on dissolved arsenic and manganese concentrations in background locations in the D/U Reach. This report presents the porewater chemistry results and discusses uncertainties in the dataset.

### 8.1. Study Design

The 2018 PDI porewater study used small-volume peepers where dissolved arsenic and manganese diffuses across the peeper membrane and reaches equilibrium with the liquid within the vial. The porewater field sampling plan (FSP) states that samples will be considered acceptable if 80 to 100% of equilibrium has been achieved. If not, a correction factor will be used to approximate the equilibrium concentrations (AECOM and Geosyntec 2018b). A 28-day period was estimated by the Pre-RD Group to be sufficient to achieve equilibrium and was therefore selected as the deployment period.

Arsenic and manganese are sensitive to the presence and amount of oxygen—when oxygen is absent, more arsenic and manganese will be dissolved in water rather than in their solid form. Nine porewater sampling locations were selected in the D/U Reach that targeted areas where dissolved arsenic and manganese concentrations would be highest and sediment concentrations were near regional background (Oregon Department of Environmental Quality 2013). This included areas low in oxygen, high in organic carbon, near wetlands, and near areas with historic presence of methane bubbles. These selection criteria targeted sampling locations that would be biased high and not representative of unbiased background concentrations. The nine porewater sampling locations in the D/U Reach are shown in **Figure 8-1**.

### 8.2. Porewater Chemistry Results and Data Uncertainties

The measured concentrations of arsenic and manganese were detected above the ROD Table 17 groundwater CULs at all locations. The arsenic CUL is regulatory-based (Clean Water Act) while the manganese CUL is risk-based from the EPA's regional screening levels (EPA 2017a, 2016b). Both CULs are based on human health criteria. None of the porewater samples achieved the 80% equilibrium threshold during the 28-day deployment, ranging from 40 to 66% equilibrium. Therefore, in accordance with the FSP, a correction factor must be applied to approximate equilibrium concentrations. The correction factor for arsenic was not provided in the FSP or the primary literature cited in the FSP. Therefore, corrected concentrations for arsenic have not been calculated at this time. **Table 8-1** summarizes the range of concentrations (measured and estimated corrected) for arsenic and manganese, respectively.

The estimated equilibrium concentrations (using a correction factor) will be higher than the measured concentrations owing to equilibrium not being achieved during the deployment period. However, the calculated equilibrium concentrations have an unknown amount of uncertainty and represent estimates rather than actual concentrations. Additionally, the correction factor for manganese has not been validated by the EPA, and the correction factor for arsenic has not been provided at this time. Therefore, it is not possible to assess its validity nor calculate corrected arsenic concentrations.

### **8.3. Conclusions**

All the porewater samples (measured and estimated corrected concentrations) were above CULs for arsenic and manganese. However, none of the samples achieved equilibrium, and the use of a correction factor to estimate equilibrium concentrations introduces additional uncertainty to the results. Furthermore, owing to the targeted nature of the sampling locations, the 2018 PDI porewater data are not representative of unbiased background conditions and the results do not supplant the regulatory- and risk-based CULs for arsenic and manganese, respectively. Rather, these data will be useful during remedial design and remedial action to understand the range of porewater concentrations that could be expected in wetlands and other reducing environments for these two contaminants.

## **9. Fish Tracking Evaluations**

At Portland Harbor, subsistence, recreational, and tribal fishers catch and consume different species of both resident (e.g., smallmouth bass) and non-resident (e.g., salmon) fish. Resident predator fish, such as smallmouth bass, present the greatest exposure risk to the site COCs owing to their small home ranges and higher trophic level (EPA 2016a). The PDI acoustic fish tracking study provides an update on previous research and will be used to better understand the life history of smallmouth bass in the site. This section summarizes the development of the exposure units in the site human health risk assessment (HHRA) and evaluates the first 9 months of acoustic fish tracking data collected by the Pre-RD Group.

### **9.1. Smallmouth Bass Home Ranges and Risk Assessment Exposure Units**

Previous studies on the home ranges, movement, and diet of resident fish in the lower Willamette River were conducted by the Oregon Department of Fish and Wildlife (ODFW) from 2000 to 2003 (Pribyl et al. 2004). These studies found that smallmouth bass preferred nearshore environments and moved more from their release location in the first month (median = 1.5 miles) relative to the rest of the study period (median = 0.25 miles). They also take spawning migrations and winter in offshore areas of deeper water and typically are not feeding during these time periods. Research has shown that bioaccumulation in fish tissue is associated with dietary uptake of contaminated food particles and aqueous uptake of dissolved contaminants (Streit 1988). Most of the site risk associated with human consumption of fish is due to PCBs and dioxins/furans, which are organic compounds that do not readily dissolve in water and therefore bioaccumulate primarily from dietary uptake (EPA 2016a; ATSDR 2019). As smallmouth bass predominantly feed and spend their time in small nearshore home ranges, their greatest exposure to contaminants occurs in these locations and at this spatial scale.

Based on these results, the site HHRA assumed a 1-river mile home range for smallmouth bass and evaluated resident fish consumption risk at 1-mile and sitewide scales with different fishing rates to account for a multi-species diet and variability in fishing behavior (EPA 2016a). Smallmouth bass were used as a surrogate for the other resident fish species (common carp, brown bullhead, and black crappie) in the 1-mile exposure unit evaluation as this was the only species where samples were composited on a 1-mile scale. Additionally, the other resident fish species are estimated to have larger home ranges and are less specifically associated with small-scale feeding stations. However, the tissue concentrations vary between species with common carp having PCB concentrations an order of magnitude higher than smallmouth bass. Owing to these assumptions, an uncertainty analysis was performed as part of the HHRA and concluded that the use of the 1-mile exposure unit for smallmouth bass and the multi-species diet would neither over nor underestimate risk and should be protective of human health (EPA 2016a).

Additionally, direct contact with in-water sediment is a potentially complete pathway for fishers and was assessed at 0.5-mile and sitewide scales. The HHRA found that human health risks vary spatially throughout the site with the highest risks at small scales in areas of elevated sediment and fish tissue COC concentrations. Therefore, the human health risk-based sediment CULs and fish tissue target levels selected in the ROD are based on the exposure risks at the 0.5-mile, 1-mile, and/or sitewide scales and represent concentrations that ought to be protective for all potentially exposed people (EPA 2017a).

## 9.2. PDI Acoustic Fish Tracking Study Design

The PDI acoustic fish tracking study was conducted over a 12-month period from May 2018 to May 2019. It was designed to track the presence and locations of 40 smallmouth bass throughout the site using acoustic receivers and implanted acoustic tags. The receivers were deployed to provide estimated locations of the tagged fish in three target areas of the site: Willamette Cove (RM 6.5E), Swan Island Lagoon ([SIL]; RM 8E), and RM 11.5E. This was done by deploying four or five acoustic receivers in a high-resolution array that would have the ability to triangulate a fish's location in the river and calculate an estimate of its location. Throughout the remainder of the site, a series of eight gates were deployed downstream (RM 1.9 and MC), upstream (RM 11.8), and at 1-river mile intervals in the middle (RMs 5, 6, 7, 8, and 9). These gates can detect when a tagged fish swims by and provide a general river mile location of the fish at a point in time but are unable to estimate its actual location.

Detection data at a gate or array was continuously recorded during the 12-month study period as tagged fish were within range, and the data were downloaded after 3, 6, 9, and 12 months. This differs from the previous fish tracking studies conducted in 2000 to 2003, which relied on opportunistic tracking during 1 to 10 days per month. However, owing to the fixed locations of the receivers (compared with the active telemetry tracking performed during the previous studies) and the receivers' detection range limitations, tagged fish could not be detected at distances greater than 1,300 feet in quiescent areas like Willamette Cove or 800 feet in noisier areas like RM 11.5E (AECOM and Geosyntec 2018a). This led to gaps in the detection histories of individual fish despite the continual tracking by the acoustic receivers.

Alternatively, the receiver gates at RMs 6 and 7, RMs 8 and 9, and RM 11.8 are potentially within range of the arrays at Willamette Cove, SIL, and RM 11.5E, respectively. This leads to fish being simultaneously detected at a gate and an array but is not necessarily indicative of fish movement between these locations. A future evaluation not covered in this report would be to define the uncertainty bounds of the receivers (i.e., range finding). This would allow for a more reliable interpretation of detection history when a fish is being simultaneously detected by gate(s) and array(s) but not positioned. The locations of the acoustic receivers are shown in **Figure 9-1**.

## 9.3. Smallmouth Bass Movement Evaluations

Data were downloaded from the receivers at 3, 6, 9, and 12 months during the study period, with fewer fish than the original 40 detected at each subsequent download event owing to fish mortality, fish leaving the study area, or fish within the study area but outside of the receivers' range (i.e., head of SIL, between MC and RM 5, between RM 9 and RM 11.5E array). As no active telemetry occurred during this study, it is not possible to know the locations of the fish that were not detected by one or more receivers, resulting in data gaps in the movement history. **Table 9-1** summarizes fish detections and available data throughout the study period.

Based on their movement characteristics, the 40 smallmouth bass detected at the beginning of the study were grouped into three qualitative movement categories. Owing to the limitations of the dataset, these categories represent broad patterns of movement and are not meant to be statistically rigorous. Rather, they provide a general classification for movement within the study area. These categories and the number of fish in each are summarized in **Table 9-2**.

Most of the tagged fish can be classified as predominantly stationary ( $n = 20$ ); these fish do not leave their release location of Willamette Cove (RM 6.5E), SIL (RM 8E), or RM 11.5E. Sixteen of the fish exhibited movement behavior, with extended periods of little movement to short-duration, long-distance movement episodes. Of the 16 fish that had some movement, 13 spent 2 months or greater during the summer (post-spawning) in the same general location, likely its feeding station during this period. One of the remaining three fish in this category may be in the head of SIL while the other two left the study area either downstream or upstream. This is consistent with the movement behavior measured by ODFW from 2000 to 2003 (Pribyl et al. 2004). Four fish exhibited long-distance movement episodes and/or left the study area shortly after being released. Approximately 1 month of data or less are available to draw conclusions about these fish. Of these four fish, two were released in the RM 11.5E array and may have been located not far upstream but out of range of the gate at RM 11.8. There do not appear to be statistical differences in the fish weight or length between the qualitative movement categories (**Figure 9-2**). Daily detection summary plots (gates and arrays) are included for each tagged smallmouth bass in **Figures 9-3a** through **9-3an**. Estimated fish location maps (arrays only) are also provided for the fish with sufficient data in **Figures 9-4a** through **9-4ad**.

There are potential aberrations in the dataset. These include the following:

- Fish detections at gate(s)/array(s) without prior detection at the adjacent upstream or downstream gate/array
- Estimated coordinate locations that appear on land an unreasonable distance from the river bank

These aberrations do not appear to have major consequences on the quality or integrity of the data but suggest that some fish detections and coordinate locations should be interpreted as estimates. Additionally, active telemetry did not occur during this study to attempt to locate undetected fish or validate the fish movement being detected by the deployed receivers. To the EPA's knowledge, only one fish was inadvertently caught during the fish tracking study. This fish was euthanized during the PDI fish tissue sampling before being properly identified. Until its incidental catch in August 2018, this fish was in Willamette Cove and has been classified as predominantly stationary for these evaluations.

## **9.4. Conclusions**

Of the 40 fish investigated during the fish tracking study, 50% exhibited little movement within or outside of their release area while an additional 35% spent 2 months or greater during the summer and fall (post-spawning) in the same general location after moving from their initial release point. Two fish (5%) that left the study area were initially released at the RM 11.5E array and may have been located just upstream of the RM 11.8 gate. These results suggest that up to 90% of the tagged smallmouth bass spent most of their feeding time in the same relatively small area, consistent with the findings of Pribyl et al. (2004). Therefore, the 2018 acoustic fish tracking study reinforces the home range estimates from the previous ODFW study and provides additional justification for the 0.5- and 1-mile exposure units in the ROD.

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## 10. Fish Tissue and Sediment Relationship at Different Averaging Radii

The *PDI Evaluation Report* (AECOM and Geosyntec 2019a) and an August 28, 2019 memorandum (Integral Consulting, Inc. [Integral] 2019a) prepared by the Pre-RD Group suggest that contaminant concentrations in fish tissue are unrelated to contaminant concentrations in sediment. In their analysis, sediment data were paired with fish tissue data by averaging over relatively large areas, including data from both sides of the river irrespective of the location where the fish were captured, or on the other extreme, averaging sediment samples within 100 feet. To evaluate the approaches used by the Pre-RD Group, the EPA developed a series of maps showing interpolated sediment concentrations and fish tissue concentrations throughout the site and performed statistical analyses linking contaminants in fish tissue to those in sediment. The site maps were developed for the five focused sediment COCs that are bioaccumulative organic chemicals and show that the highest sediment concentrations for a COC also have elevated fish tissue concentrations. The statistical analyses were based on averaging of sediment sample locations over spatial scales ranging from within 100 feet up to 5,280 feet (i.e., within 1 mile of the capture location) and restricted averaging to the side of the river, where the relevant fish were captured (east, west, or navigation channel). The statistical models fit to the data treated fish tissue contaminant concentrations as a log-log relationship with sediment contaminant concentration, fish lipid content, and sediment organic carbon content. The methods and results of the analysis are described in the following sections.

### 10.1. Methods

Simple bioaccumulation models for organic contaminants usually relate lipid normalized contaminants in fish to organic carbon normalized contaminants in sediment. This normalization is intended to reflect the differential accumulation rates as they vary as a function of the ratio of lipid in tissue to organic carbon in sediment. Fish exposed to contaminants in sediments with lower organic carbon content are expected to exhibit proportionally higher body burden relative to fish exposed to contaminants in sediments with higher organic carbon content. Similarly, for fish exposed equally to contaminants, wet weight contaminant concentrations in tissue are expected to be proportionally greater for fish with greater lipid content. The biota to sediment accumulation factor (BSAF) reflecting these relationships is usually given by the following equation:

$$\frac{C_f/f_L}{C_s/f_{oc}} = BSAF \quad \text{Equation 1}$$

Where

$C_f$  = the contaminant concentration in fish tissue

$C_s$  = the contaminant concentration in sediment

$f_L$  = the fraction of lipid in the fish specimen

$f_{oc}$  = the fraction of organic carbon in sediment

#### 10.1.1. Generalization of BSAF

Hebert and Keenleyside (1995) found that normalized relationships have some implicit assumptions of a linear regression through the origin that may not hold and further showed that the relationships can be masked when either tissue contaminant concentrations are independent of

lipid content or when organic carbon is independent of sediment contaminant concentrations. Generally, they recommended restructuring normalized relationships as regression models and testing for the requisite relationships before arbitrarily normalizing measurements. The EPA followed this advice by noting that the BSAF relationship can be decomposed into a multiple linear regression model that provides a framework for testing the contribution of each term in the BSAF model for relevance in predicting fish tissue concentrations from sediment exposure. By taking natural logarithms of both sides of the BSAF relationship, the equation can be expressed in a more general form:

$$LN(C_f) = LN(BSAF) + \beta_1 LN(f_L) + \beta_2 LN(f_{OC}) + \beta_3 LN(C_s) \quad \text{Equation 2}$$

When  $\beta_1 = -\beta_2 = \beta_3 = 1.0$ , this multiple regression simplifies to the usual BSAF equation. However, when coefficients differ from 1.0, the usual BSAF relationship is inappropriate because the relationship may be nonlinear. When  $\beta_1 = 0$  or  $\beta_2 = 0$ , accumulation should be expressed independently of fish lipid content or sediment organic carbon content, respectively. Finally, a relationship between fish tissue contaminant concentration and sediment contaminant concentration is evaluated by testing the null hypothesis ( $H_0: \beta_3 = 0$ ). Reorganizing Equation 2, the accumulation relationship is similar in form to the BSAF, with allowance for nonlinear relationships and potential that some variables may not be informative in estimating fish tissue concentrations from exposure to contaminated sediments.

$$C_f = (BSAF) \times f_L^{\beta_1} \times f_{OC}^{\beta_2} \times C_s^{\beta_3} \quad \text{Equation 3}$$

In this form, concentrations in fish tissue are proportional to powers of lipid, organic carbon, and sediment concentrations. The BSAF is the special case of this relationship found by setting  $\beta_1 = -\beta_2 = \beta_3 = 1.0$  as described above.

The EPA fit Equation 2 to data using the maximum likelihood estimation, assuming that the log of mean fish tissue concentration is linear in logarithm of fraction lipid, fraction organic carbon, and contaminant concentration in sediment. Maximum likelihood estimation is needed for these models because standard least squares fitting would result in biased estimates of the mean and would more tightly constrain the residuals to be lognormally distributed.

#### **10.1.2. Scale of Fish to Sediment Relationships**

The sediments to which fish are exposed are uncertain because the life history of the fish that were sampled is unknown. The EPA averaged sediment samples over a range of distances from the capture location from 100 feet up to 5,280 feet (i.e., 1-mile radius or 2 river miles overall) in increments of 100 feet and tested the null hypothesis ( $H_0: \beta_3 = 0$ ) for each scale of averaging. The EPA then plotted the statistical level of significance against scale of averaging to evaluate what scale of averaging provided the strongest evidence of a relationship between fish tissue concentration and sediment concentration. The EPA applied this approach to the five focused sediment COCs that are bioaccumulative organic contaminants at the site, and the results are presented in the following section. Finally, the EPA tabulated the model results for the scale of averaging that was best for each model.

## 10.2. Results

The results from the statistical analyses for the five bioaccumulative organic focused sediment COCs is included in the subsections below.

### 10.2.1. Association Between Fish and Sediment Contaminant Concentrations

For all five organic contaminants, the EPA found statistically significant positive associations between fish tissue concentrations and average sediment concentrations (i.e.,  $\beta_3 > 0$ ;  $p < 0.05$ ) when sediments included for averaging were restricted to the side of the river where the fish was captured and when the radius of averaging was relatively small—on the order of 100 to 600 feet (**Figure 10-1**). The strongest relationships were found for averaging areas less than 500 feet for all five contaminants, suggesting that fish exposures were dominated by sediment concentrations proximal to capture locations as opposed to averages representing larger exposure areas.

Tissue concentrations of 1,2,3,7,8-PeCDD were unrelated to sediment contaminant concentrations when averaging radii were greater than approximately 600 feet ( $p > 0.05$ ). For the other four organic contaminants, fish tissue contaminant concentrations were also associated with sediment concentrations averaged over larger sediment averaging areas although strength of relationships decreased with increasing averaging area size.

### 10.2.2. Best Models

The models fit to the data provide a framework to explain contaminant variability in tissue due to variation in exposure to contaminants in sediment and covariation with lipid content in fish and organic carbon content in sediment. Both lipid and organic carbon are expected to influence accumulation of organic contaminants. This analysis provided a way to account for these components of variation, which in general results in a more precise estimation of the relationship between tissue and sediment contaminant concentrations. For the spatial averaging scales found to result in the strongest relationships, the EPA summarized the full models in **Tables 10-1** through **10-5**. For all five organic contaminants, tissue concentrations were independent of lipid content, indicating that traditional lipid normalization is unnecessary and could tend to mask relationships between tissue and sediment concentrations based on traditional BSAF-like analyses or linear regressions between normalized concentrations (Hebert and Keenleyside 1995).

#### 10.2.2.1. 1,2,3,7,8-PeCDD

Concentrations of 1,2,3,7,8-PeCDD in fish tissue were positively associated ( $\beta_3 = 0.46$ ;  $p < 0.001$ ) with sediment concentrations and negatively associated with organic carbon content ( $\beta_2 = -0.39$ ;  $p = 0.03$ ). The coefficient for sediment differed from 1.0, and that for organic carbon differed from -1.0, indicating that contaminant accumulation in tissue is nonlinear in both sediment concentration and organic carbon in contrast to the usual assumptions of linearity with the BSAF model. The coefficient on sediment ( $\beta_3$ ) is less than 1.0, indicating that the ratio of fish to sediment concentrations declines with increasing contaminant concentrations. For organic carbon, because the coefficient is greater than -1.0 (as assumed in the BSAF model), accumulation per unit of organic carbon is less than would be predicted by the BSAF model.

#### 10.2.2.2. 2,3,4,7,8-PeCDF

Concentrations of 2,3,4,7,8-PeCDF in fish tissue were positively associated with sediment concentrations ( $\beta_3 = 0.67$ ;  $p < 0.001$ ) and negatively associated with organic carbon content ( $\beta_2 =$

–0.46;  $p = 0.008$ ). As with 1,2,3,7,8-PeCDD, both coefficients differed from 1.0 and –1.0, respectively, indicating a nonlinear relationship between tissue concentrations, sediment concentrations, and organic carbon. Again, these data indicate that accumulation is less at higher sediment contaminant concentrations than would be predicted with a linear accumulation model. This nonlinearity in organic carbon also indicates that changes in accumulation are less responsive to changes in organic carbon than a linear model would predict.

#### **10.2.2.3. 2,3,7,8-TCDD**

Tissue 2,3,7,8-TCDD concentration was positively associated with sediment concentration content ( $\beta_3 = 0.16$ ;  $p = 0.005$ ) but unrelated to both fish lipid ( $p = 0.34$ ) and sediment organic carbon ( $p = 0.28$ ). As with the other organic contaminants, accumulation is lower at higher sediment concentrations than would be predicted by a linear regression or BSAF-based accumulation model.

#### **10.2.2.4. Total DDx**

Total DDx concentrations in fish tissue were positively associated with DDx in sediment ( $\beta_3 = 0.23$ ;  $p = 0.002$ ) but unrelated to fish lipid content ( $p = 0.68$ ) or sediment organic carbon ( $p = 0.67$ ).

#### **10.2.2.5. Total PCBs**

As with total DDx, fish tissue total PCB concentrations were positively associated with sediment total PCB concentrations ( $\beta_3 = 0.23$ ;  $p = 0.002$ ) but unrelated to fish lipid and sediment organic carbon. As with the other organic contaminants, accumulation from sediment is lower at higher levels of sediment PCBs than would be predicted with a linear regression or BSAF accumulation model.

### **10.2.3. Spatial Analysis**

Maps were developed for total PCBs; total DDx; 2,3,7,8-TCDD; 1,2,3,7,8-PeCDD; 2,3,4,7,8-PeCDF; and total PAHs showing interpolated surface sediment concentrations from the combined RI/FS and PDI/BL data along with the smallmouth bass sampling locations. Visual inspection of **Figures 10-2** and **10-3** show a strong relationship between the PDI fish tissue concentrations and the interpolated sediment concentrations for total PCBs and total DDx, respectively. This relationship is also present in the dioxins/furans (**Figures 10-4** through **10-6**) although less easily discerned visually. The PDI fish tissue data do not contain results for total PAHs; therefore, **Figure 10-7** shows the interpolated sediment concentrations and the 2007 RI fish tissue composite samples.

## **10.3. Discussion**

These analyses indicate that bioaccumulative contaminants in fish are best described by the concentration in sediments near where they were captured. This contrasts with the conclusions drawn in the *PDI Evaluation Report* (AECOM and Geosyntec 2019a) based on the contaminant data and acoustic fish tracking study. The acoustic fish tracking data suggest that the home ranges of smallmouth bass within the lower Willamette River appear to be larger than the optimal sediment averaging areas, but the scale of sediment averaging is smaller than the home range sizes. This at first may seem contradictory; however, there are several reasons why contaminant exposure areas may not align fully with the size of an organism's home range.

Visual inspection of the capture locations shows that smallmouth bass that were caught by hook and line (i.e., while they were feeding) were consistently along the shallower shoals where

sediment contaminant concentrations are highest as opposed to within the deeper navigation channel where contaminants are generally lower. The acoustic fish tracking data indicate that smallmouth bass home ranges were generally larger than the 100-to 600-foot-radius sediment averaging areas evaluated in the statistical analyses, including time spent in deeper water in the navigation channel. These observations are consistent with previous studies on smallmouth bass behavior in the lower Willamette River (Pribyl et al. 2004), which show that the fish generally cycle between feeding in shallow waters that have more contaminated sediments and retreating to deeper adjacent waters that have more consistent temperatures year-round. With such a cycle and with a substantial component of tissue contaminant burden likely originating from dietary exposure rather than aqueous exposure (Streit 1988), one would expect the sediments in feeding areas on the shoals where food items are exposed to better reflect tissue concentrations in the fish—consistent with the findings that intermediate to small scales of averaging provide the best predictor of tissue concentrations.

Developing empirical relationships in contaminant concentrations in tissue and sediment presents difficulties: (1) the sediments which provide a source directly and indirectly to fish exposure are unknown; (2) measurements of average concentration in these sediments even when known are variable; and (3) the degree to which exposure is apportioned between dietary uptake, direct exposure to contaminated sediments, and aqueous uptake are also unknown. Because of these uncertainties, it is expected that relationships estimated from sample data would be uncertain. In general, uncertainty in measurements tends to dampen regression relationships, which would result in accumulation functions that underpredict tissue concentrations based on the available sediment data. Nonetheless, statistically identifiable empirical relationships between sediment and fish tissue contaminant levels have been found for the five focused sediment COCs that are known bioaccumulative organic compounds. Because of the uncertainties described above, the EPA expects that these empirical estimates approximate the actual underlying mechanistic relationships that have been studied and developed through controlled experimentation and models and underpredict the resulting tissue concentrations.

The *PDI Evaluation Report* (AECOM and Geosyntec 2019a) argues that the mechanistic food web model (FWM) used to develop sediment CULs at the site overpredicts tissue accumulation because the empirical models in the *PDI Evaluation Report* indicate less accumulation. The empirical relationships summarized in this report and the *PDI Evaluation Report* (AECOM and Geosyntec 2019a) are estimated and are insufficient to replace the mechanistic FWM used to develop sediment CULs at the site. The mechanistic FWM is based on extensive peer-reviewed literature that identifies and quantifies the underlying mechanisms driving bioaccumulation. Furthermore, the mechanistic FWM has been empirically calibrated to site conditions to account for uncertainty in the data. The overall conclusion is that bioaccumulative contaminants in fish tissue are associated with contaminant concentrations in sediment near the areas where these fish were caught. Smallmouth bass contaminant exposure is driven primarily through dietary uptake, and therefore the sediment contaminant concentrations in foraging areas along the shoals are most representative of exposure.

## **10.4. Conclusions**

These results indicate four primary findings:

- 1) In contrast to the findings in the *PDI Evaluation Report* (AECOM and Geosyntec 2019a), all five bioaccumulative focused sediment COCs in tissue were positively associated with contaminant concentrations in sediment provided that averaging was restricted to the side of the river where fish were collected.
- 2) The strongest relationships for all contaminants were identified when fish tissue concentrations were paired with sediment concentrations found near fish capture locations.
- 3) Analyses based on lipid and organic carbon normalized data should be interpreted cautiously and may be counterproductive because they tend to mask relationships between tissue and sediment concentrations.
- 4) The empirical relationships between fish tissue and sediment concentrations modeled in this report and in the *PDI Evaluation Report* (AECOM and Geosyntec 2019a) are an estimate and, in contrast to a mechanistic FWM, do not incorporate empirical calibration to site conditions or underlying mechanisms driving bioaccumulation. As such, the relationships are informative but insufficient to replace the mechanistic FWM used to develop sediment CULs at the site.

## 11. Differences in RI/FS and PDI/BL Study Designs

The EPA disagrees with the conclusions in Integral (2019b) that the differences in study design between the RI/FS and PDI/BL programs do not influence temporal comparisons of SWACs between the two datasets. Consistent with the EPA's comments on the *PDI Evaluation Report* (AECOM and Geosyntec 2019a), the EPA considers the temporal comparisons of RI/FS and PDI/BL SWACs to be estimates for the following reasons.

The PDI/BL SRS surface sediment samples were randomly placed within grid cells of known area for the express purpose of developing a statistically unbiased baseline dataset for the long-term monitoring program. The data use objectives pertaining to the SRS surface sediment samples are listed in Section 1.3 of the PDI work plan and are as follows (Geosyntec 2017):

1. Implement investigation baseline sampling to update existing sitewide data
2. Gather data to be used as part of baseline dataset for future long-term monitoring

The SRS surface sediment sample grid cells are distributed throughout the site from RMs 1.9 to 11.8. The placement of the grid cells and SRS surface sediment samples was done to generate sitewide coverage for the unbiased baseline dataset. The stratified random sampling design allows the data to have a known statistical bias that can be calculated. This bias can be adjusted for based on the spatial weighting of the grid cell areas and the number of samples in the shoals versus the navigation channel, leading to the calculation of unbiased SWACs. Assuming the stratified random sampling design is repeated as part of the long-term monitoring program, statistically robust temporal rates of change can be calculated from these unbiased SWACs.

The RI/FS surface sediment data consist of samples that were collected for delineating the nature and extent of contamination and for developing the baseline risk assessments as detailed in Section A5.2 and Appendix A of the *Round 2 Quality Assurance Project Plan* (Integral and Windward Environmental, LLC 2004). As stated in Integral (2019b), the sitewide RI/FS surface sediment samples were not located in a stratified random grid but rather were in areas of low sample density. The RI/FS sampling program did not contain a data use objective of collecting an unbiased baseline dataset. Therefore, these data represent a non-random and partially random sampling scheme as defined in *Geostatistical Sampling and Evaluation Guidance for Soils and Solid Media* (EPA 1996), which will likely produce a biased estimate of the mean.

While spatial weighting of datasets with unknown bias (e.g., RI/FS data) using interpolation methods such as natural neighbor and Thiessen polygons can reduce the bias when calculating means, the resulting SWACs still contain some measure of uncertainty (Kern 2009). Uncertainty in the RI/FS data was assessed during the FS and is summarized in FS Appendix I (EPA 2016b). Uncertainty in the natural neighbor and Thiessen polygon interpolated PDI/BL data was not presented in the *PDI Evaluation Report* (AECOM and Geosyntec 2019a) and cannot be assessed at this time.

Assessing temporal change in sediment concentrations is necessary to measure the progress of the remedy toward attaining site CULs. However, owing to the different data use objectives and study designs, unknown bias in the RI/FS data, and uncertainties with interpolation-based SWACs, temporal comparisons between the RI/FS and PDI/BL data should be considered estimates. These

estimates can provide a qualitative understanding of how contaminant concentrations in surface sediment have changed since the RI/FS data were collected. Statistically robust rates of temporal change can be calculated by comparing SWACs developed from the baseline SRS surface sediment samples to future long-term monitoring sample data that replicate the stratified random design.

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## Tables

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## Figures

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